



Quantification of tillage and landscape effects on soil carbon in small Iowa watersheds

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Abstract

Knowledge of the long-term effects of tillage on soil organic carbon is important to our understanding of sustainable agricultural systems and global carbon cycles. In landscapes susceptible to erosion, tillage can exacerbate losses of soil and C by increasing erodibility and stimulating microbial respiration. We measured long-term changes in soil carbon and soil loss in three small watersheds located in southwest Iowa, USA. The following soil series were formed on deep loess hills: Ida and Dow (Typic Udorthents), Napier and Kennebec (Cumulic Hapludolls) and Monona (Typic Hapludolls). All watersheds were cropped to continuous corn (*Zea mays* L.) and two were moldboard plowed and disk tilled while the third was ridge-tilled. The ridge-tillage system had greater C contents in the surface soil than the disk tillage soils, but ridge-tillage was not different from the conventional tillage in carbon retention over time. The ridge-tillage system, however, was more effective in retaining soil within the watershed. Microbial respiration by soil microorganisms accounted for 97% of the carbon loss in the ridge-tilled watershed compared to carbon loss in eroded sediment (3%). Terrain analysis was used to segment the landscape into landform elements. Less total carbon was present in the soil profiles of backslope elements than in footslope or toeslope elements, reflecting the combined effects of soil erosion and deposition within the watersheds. Profile C content was also positively correlated with the wetness index, a compound topographic attribute, that identifies areas of the landscape where runoff water and sediment accumulate.

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1. Introduction

Agricultural practices affect the sustainability of crop production systems and impact water quality in many regions of the USA and elsewhere. Increasingly, watersheds are being considered as the fundamental unit for evaluating linkages between agricultural practices and water quality. Implicit in the watershed-scale evaluation of agricultural practices is the influence of soil and terrain on the processes controlling crop pro-

duction and losses of sediment, nutrients and pesticides into surface and ground water. Soil organic carbon, as total organic C or biologically active components (particulate organic matter, microbial biomass) and biological processes (respiration, nutrient mineralization) are also thought to be important indicators of soil quality and agricultural sustainability (Doran et al., 1996; Seybold et al., 1997). Several studies address assessments of soil quality at regional or national scales (Brejda et al., 2000; Sparling and Schipper, 2002). The watershed and landscape scale is smaller in scope than regional or national assessments, but larger than the field plot that has been most often used for soil quality studies.

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Tillage can modify many soil properties, including organic matter content, microbial biomass, and aggregation (Cambardella and Elliott, 1992, 1993; Frey et al., 1999). Tillage-induced effects on basic soil properties can then become evident at the watershed-scale by affecting processes such as infiltration, runoff and sediment loss (Edwards et al., 1993; Rhoton et al., 2002). Despite the long history of research on tillage systems there are relatively few reports document tillage effects on both soil properties and erosional processes, particularly at the watershed-scale.

The objective of this research was to evaluate the long-term effect of ridge-till or conventional (moldboard plowing or disking) tillage on soil carbon dynamics within three, field-sized, watersheds in the loess hills of southwest Iowa. Specifically, we examine long-term changes in soil organic carbon at the watershed-scale and the variation in profile soil carbon among different landforms within the watersheds.

2. Methods

2.1. Watershed description and sampling

The three watersheds used in this research are near Treynor, IA in the deep loess hills that occupy western Iowa and northwestern Missouri. Monona silt loams (Haplic Phaeozems in FAO classification) are found in the summit and shoulder landform elements, Ida and Dow silt loams (Calcaric Regosols) in backslopes, and Napier and Kennebec silt loams (Cumulic–Haplic Phaeozems) in the footslope and toeslope elements. The watersheds range from 27.6 to 43.3 ha in size and each includes a small perennial stream fed by baseflow. Surface runoff and baseflow were monitored at weirs at the base of each watershed (Fig. 1). Automated sediment samplers located approximately 50–125 m above the weirs were used to collect runoff sediment samples. The location of these samplers was such that they could effectively sample the concentrated runoff leaving the watershed coming directly from the fields via overland flow, while eliminating the contribution of bank and gully erosion which occurs in the zone between these samplers and the weirs. Sediment content of runoff water combined with runoff flow is reported in this paper as an estimate

of soil (sediment) loss from the cropped areas of the watersheds.

From 1972 until 1995 watersheds W1 and W2 were cropped to continuous corn (*Zea mays*, L.) with moldboard plowing until the early 1980s and disk tillage thereafter. Ridge-tillage was used since 1972 in watershed W3 (Karlen et al., 1999). Ridge-tillage is a contour-based system that involves planting into a ridge, then using one or two postemergence disk tillage operations to rebuild the ridge and to control weeds. For the 8 years preceding conversion to ridge-till (1964–1971), watershed W3 was a brome grass (*Bromus inermis* L.) pasture, while watersheds W1 and W2 were cropped with corn.

Plots established for the measurement of corn yield (Karlen et al., 1999) were used as a framework for post-harvest soil sampling in 1994 and 1995. Soil samples consisting of two cores, 7.6 cm in diameter and 15 cm deep, were taken at approximately 25 m intervals along transects between yield plots that had been established in the 1964–1972 period. In general, these transects begin on the hilltops or shoulders and end on the footslopes (Fig. 1). In addition, smaller cores of a known volume were taken by hand to the same depth for the measurement of bulk density. Soil samples were placed in insulated chests in the field and stored at 4 °C until analysis. Sampling locations were marked and referenced by survey methods to benchmark locations within the watersheds. These data were later converted into a georeferenced database using Arc/Info¹ GIS software (Version 7.1.1, Environmental Systems Research Institute, Redlands, CA, USA).

At every third or fourth sampling location on each transect a 120 cm deep soil core was taken with a truck-mounted probe. These were divided into 15 cm increments and analyzed for organic carbon, pH and carbonates. These data were used to calculate the profile organic carbon content to a 90 cm depth using a value of 1.3 g soil cm⁻³ for subsurface bulk density. For the 0–15 cm increment we utilized bulk density calculated from sampling in the field. The subsurface bulk density estimate was based on measurements

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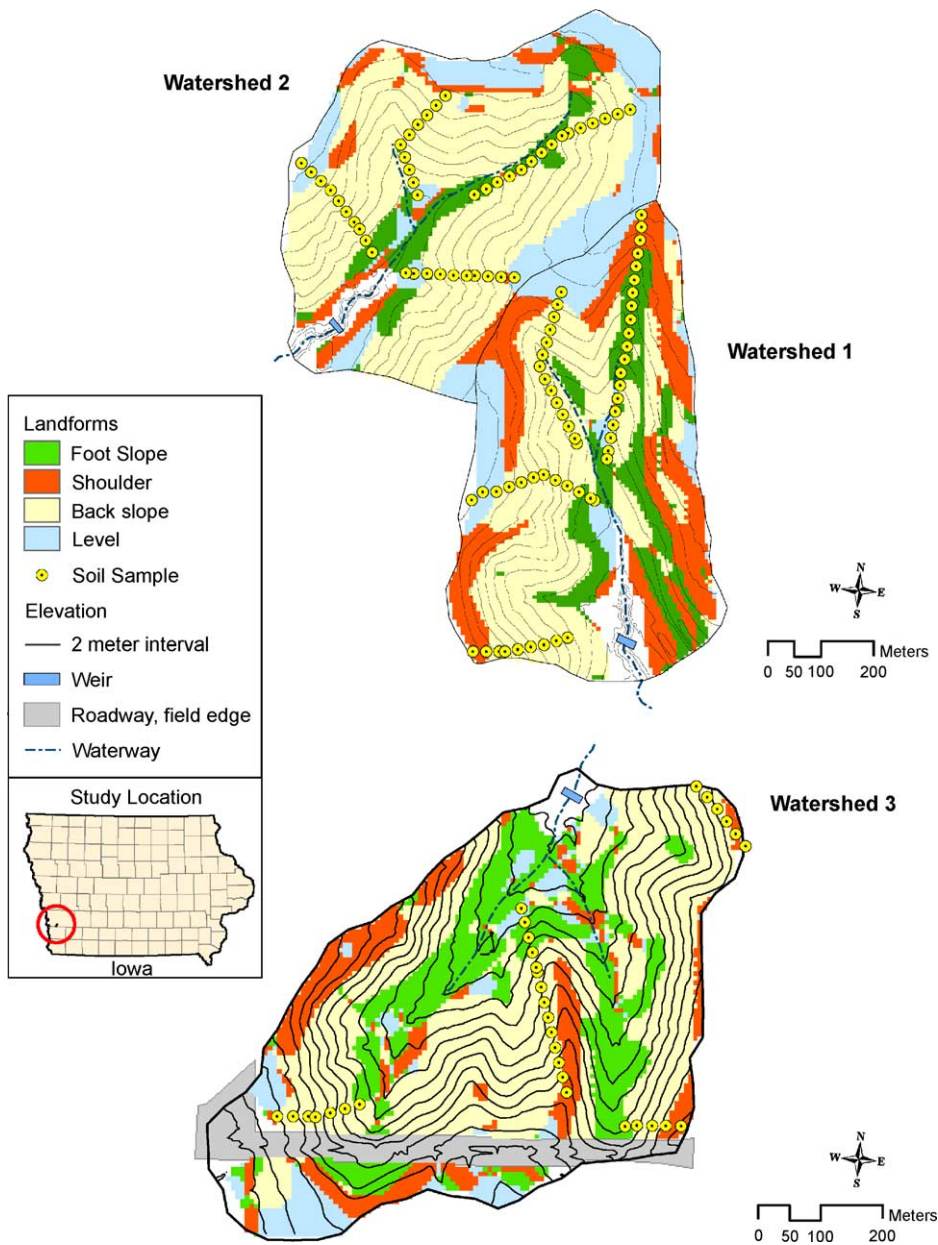


Fig. 1. Distribution of landform elements in watersheds and the sampling transects. The sampling locations represent $7.5 \text{ m} \times 7.5 \text{ m}$ cells on the DEM and are not drawn to scale. The map has been produced using a despeckling routine to produce a general representation of the landform elements. We subdivided the level landform into summit (level elements at the top of the slopes) and footslopes (level elements at the base of the slopes) on the basis of elevation.

from a smaller number of cores from these watersheds (S.L. Logsdon, personal communication) and is close to the bulk densities measured in watershed W1 by Harden et al. (2002).

Soil samples for C analysis were passed through a 2 mm sieve, air-dried, milled to a fine powder, and stored at room temperature. Measurement of organic C was accomplished by combustion using a Carlo Erba NA1500 NCS elemental analyzer, with pretreatment with acid to remove carbonates. Results obtained on dry weight were converted to an area basis using the bulk density data.

2.2. Terrain analysis and statistics

In order to examine the relationship among soil properties, terrain attributes, and landscape features, digital elevation models (DEM) were constructed from aerial photographs. The primary and secondary terrain attributes and landform elements were derived from the DEM and these attributes were spatially linked to the soil sampling locations through the GIS.

The digital elevation data were derived from processed stereo-triplets of aerial photography dated 28 April 1997 at a 1:8000-scale imagery. Ground control markers (1 m²) placed at locations on the perimeter of the watersheds prior to the overflight for use during rectification of the aerial photography. Coordinates for the ground control markers were generated using roving global positioning system (GPS) receivers and a base-station GPS receiver to differentially correct the coordinates. Calculated root-mean-square error (RMSE) for the base-station location was 0.083 m circular horizontal error and 0.062 m vertical error. The ground control locations and experimental data were referenced to the Universal Transverse Mercator (UTM) projection (Zone 15, North American Datum of 1983). Photographs were processed using a ZEISS P3 analytical stereo plotter to produce a digital terrain model (DTM) (personal communication, W. Ertz, Aerial Services Inc., Cedar Falls, IA). Ground surface locations and associated elevations were estimated using a 15 m spacing between DTM points. In addition, breaklines were delineated where rapid changes in the land surface occurred. Location coordinates and elevation values were estimated along the breaklines approximately every 3 m.

The DTM was the source data for generating a DEM consisting of 7.5 m × 7.5 m size grid cells for each watershed using the Arc/Info TOPOGRID. For each cell, slope, curvature, profile curvature, and plan curvature were calculated using the Arc/Info CURVATURE command. Estimates for specific catchment area was generated using TAPES-G software (Terrain Analysis Programs for Environmental Sciences—Grid Version, Centre for Resource and Environmental Studies, The Australian National University, Canberra, AU) that uses a modified deterministic-8 node algorithm which allows flow dispersion and catchment spreading to be represented (Moore, 1996). We classified cells with negative profile curvature (<−0.1/100 m) as footslopes. Shoulder slopes exhibited positive profile curvatures greater than 0.1/100 m and cells between −0.1 and 0.1/100 m profile curvature and more than 3° m^{−1} slope backslopes were classified as backslopes. Level landform elements had profile curvatures between −0.1 and 0.1/100 m and less than 3° m^{−1} slope. Level landforms elements were further subdivided based on elevation into summit elements (level uplands) which are generally positioned along the hilltops, and toeslope elements (lower level), which generally occupy positions below the footslopes.

In addition to the simple terrain attributes, three compound indexes were examined. The sediment transport index ($STI = (A_s/22.13)^{0.6} \times (\sin(\text{slope})/0.0896)$); the stream power index ($SPI = A_s \times (\tan(\text{slope}))$); the wetness index ($WI = \ln(A_s/\tan(\text{slope}))$). A_s is the specific catchment area contributing water to each cell, with units of m² area per m cell width (m² m^{−1}). Additional information on these indexes can be found in Gallant and Wilson (2000).

Statistical analysis included the non-parametric Wilcoxon rank sum/Kruskal–Wallis test (NPAR1WAY procedure, SAS) where non-normal data were found, or analysis of variance (ANOVA) with watersheds and landforms as main effects using the GLM procedure of SAS (Statistical Analysis Systems, Cary, NC, Version 6.0). Since the number of samples from the different landforms were not equal, the Type III sums of squares were used in the ANOVA. When significant differences were indicated by the Wilcoxon test or the ANOVA, means difference testing was performed using Duncan's multiple range test.

3. Results and discussion

3.1. Tillage and erosion effects on soil carbon at the watershed-scale

The amount of organic carbon in soils is governed by C inputs via primary plant production and by C losses, which are mainly through microbial respiration and erosion, both of which are influenced by tillage. Soil erosion is controlled principally by three factors: rainfall, terrain, and the agricultural management system. In this comparison, the close proximity of the three watersheds to each other minimized differences in rainfall, with an annual average of 816 mm. Also, except for tillage, the management of the three watersheds was similar in the 1972–1995 period, with continuous corn grown in each watershed. The distribution of landforms within the watersheds, as determined by our terrain analysis methods, is shown in Fig. 1. There are some differences among the watersheds with respect to their physical topography. Principally, watershed W1 is the most steeply sloping with a median slope of 5.3%, while W2 and W3 have median slopes of 4.2 and 4.8%, respectively. The steeper slopes in W1 result in slightly more water being partitioned into runoff in W1 than W2 (Kramer et al., 1999).

The change in organic matter of the surface soils (0–15 cm depth) in these watersheds over time is shown in Table 1. For the 1972 and 1984 samplings, bulk density data were not available. The estimated

bulk densities were made based on NRCS soil survey data and data from Allen (1971). We used measured bulk density data for the recent samplings to eliminate the effects of tillage prior to sampling in 1994. In 1972, the soil organic C contents differed among the watersheds, with W3 soils having greater C contents and W1 soils having the least. The elevated soil C levels in W3 reflect the 10-year period immediately preceding 1972 when this watershed was used as a pasture rather than cropped to corn or soybeans (*Glycine max* (L.) Merr.) (Karlen et al., 1999). The organic C levels now present in the soils represent approximately 50% of their original C content, based on a comparative study of soils in the southeast part of watershed W1 and the undisturbed Dinesen Prairie in the loess hills northeast of Treynor (Manies et al., 2001). The differences in soil C between watersheds W1 and W2 in 1972 may reflect the differences in slope, as our records show that the two watersheds were managed in a similar manner for the decade prior to 1972.

Significant amounts of C were lost in the 12 years following 1972 in watersheds W2 and W3. Conventional plow tillage in W2 resulted in 54% of the topsoil C being lost in this period, while 40% of the soil C was lost from the ridge-tilled soils (W3) over the same period of time. In contrast, the W1 soil organic C changed only slightly between 1972 and 1984. In the 11 years following 1984, soil C changes were much smaller in magnitude than the 1972–1984 period

Table 1
Estimated change in soil organic C levels in surface soil as affected by tillage

Watershed	Year	Mean soil organic C		
		(g kg ⁻¹) ^a	(Mg ha ⁻¹) ^b	(% of 1972)
W1—disk tillage	1972	15.0	28.1 ± 3.0	100
	1984	14.5	27.2 ± 0.9	97
	1995	15.1	25.2 ± 0.8	90
W2—disk tillage	1972	25.8	48.4 ± 4.0	100
	1984	11.9	22.3 ± 2.7	46
	1995	14.3	23.9 ± 0.7	49
W3—ridge-tillage	1972	31.2	58.5 ± 7.3	100
	1984	18.7	35.1 ± 2.1	60
	1995	18.7	29.7 ± 0.8	51

^a Organic C, determined from Walkley–Black measurements in 1972 and 1984.

^b Calculated to a depth of 15 cm assuming a bulk density of 1.25 Mg m⁻³ for all three watersheds for 1972 and 1984. Field-measured bulk densities were used for the 1995 calculation. Values are the mean and standard error.

(Table 1). This may indicate that decay-resistant forms of organic matter have become increasingly dominant, thus slowing the loss of organic C as the years of cultivation increase. Another factor that may explain the slower loss of soil C in the period after 1984 is the change from moldboard plowing to disk tillage in watersheds W1 and W2. Decreased rates of C loss over time have been noted in other studies (Paustian et al., 1997). By 1995, both watersheds W2 and W3 had lost about 40% of the soil C originally present in soil in 1972.

Crop biomass production can affect the soil C balance (Paustian et al., 1997). Mean corn grain yields for 1972–1995 were 7.7 ± 2.1 , 7.5 ± 2.5 , and $8.0 \pm 2.3 \text{ Mg ha}^{-1}$ for W1, W2, and W3, respectively (Cambardella et al., 2004). Furthermore, the average grain yield trends over time are nearly constant in these watersheds. Since above-ground and below-ground crop biomass is related to grain yield (Huggins and Fuchs, 1997), the small increase in carbon inputs to the soil from crop biomass production (estimated from grain yields) do not account for any differences in soil organic C among the watersheds.

Soil erosion from 1972 to 1995 from the ridge-tilled W3 was less than that from watersheds W1 and W2 (Fig. 2). For watersheds W1 and W2, the cumulative soil losses were 292 and 243 Mg ha^{-1} , respectively, for the 1972–1995 period. The cumulative sediment loss in watershed W3 was only 32 Mg ha^{-1} during the same time. The change in tillage from plowing to disking in the mid 1980s corresponds to a decline of 15 and 24% in the average annual runoff from watersheds W1 and W2, respectively, in the post-1984 period compared to the 1972–1984 period. Runoff also declined in watershed W3 after 1984, but only by 3%. Ridge-tillage modifies the land surface, decreasing the amount of water lost as runoff and increasing the base-flow (Kramer et al., 1999). Sediment losses from W3 are generally small in most years, with a few years responsible for the bulk of sediment loss.

The contribution of soil erosion to organic C loss was calculated from sediment loss and estimated soil organic C contents. For each watershed, average soil organic C contents (g kg^{-1}) for each year were estimated by interpolation using data in Table 1. These estimates were then combined with the total annual sediment mass measured at the base of the watersheds. This may underestimate C loss slightly due to enrich-

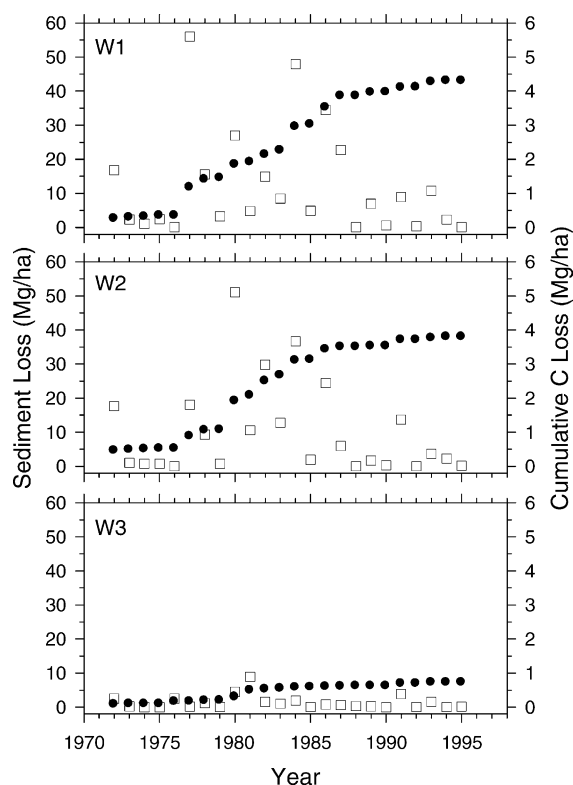


Fig. 2. Annual losses of sediment (open squares) and cumulative carbon (C) losses (circles) from watersheds in southwest Iowa. Watersheds W1 and W2 were moldboard plowed till the early 1980s and disk tilled afterwards. Watershed W3 was ridge-tilled.

ment of the soil C at the soil surface, but both the ridge-till and the conventional tillage practices prevent significant stratification of carbon within the upper 15 cm of soil. The cumulative losses of C in the sediment during 1972–1995 were estimated to be 4.3, 3.8, and 0.7 Mg C ha^{-1} for watersheds W1, W2, and W3, respectively (Fig. 2). These organic C losses are equivalent to average annual losses of 187 kg C ha^{-1} for W1 and 165 kg C ha^{-1} for W2. Cumulative C losses (0.7 Mg C ha^{-1}) in eroded sediment were much less in watershed W3 due to the ridge-tillage management. The losses of C due to erosion comprise approximately 15% of the total C loss in W2 and 2.5% of the total C loss in W3. Therefore, the bulk of C loss in W2 and W3 were due to biological oxidation of the soil C. The erosional losses of soil C from W1 and W2 are substantially greater than C losses measured in small row-cropped watersheds near Coshocton, OH (Owens

et al., 2002). In watershed W1, C loss due to erosion was greater than the net organic C loss, indicating that erosion was a significant process in this watershed. In this watershed, carbon inputs from plant production were apparently able to offset some of the C losses due to erosion. These losses represent net export from the watershed and do not represent any movement of soil within the watershed.

3.2. Soil carbon distributions within watersheds

At the watershed-scale, organic C content expressed as Mg C ha^{-1} in the surface soil (15 cm depth) sampled in 1994 and 1995, was significantly greater in the ridge-tilled soil (W3) than in the conventional tilled soils of W1 or W2 (Table 2). This result was obtained with both parametric (ANOVA) and the non-parametric Wilcoxon rank sum test. These statistical procedures also showed landform to be a significant main effect at the $P \leq 0.10$ level (Table 2), but the landform by watershed interaction was not significant for total organic C in soil profiles to a depth of 15 cm. Among the landforms, the toeslopes had significantly greater amounts of organic C, while the backslope soils had less C than the soils in other landscape elements.

The total C in soil profiles, expressed as total carbon present to a depth of 90 cm, was also evaluated using the sparser set of data obtained from the deep cores. Analysis of variance and non-parametric analyses both showed watershed and landform element effects to be statistically significant ($P \leq 0.05$). Analysis of variance also showed that the interaction of watershed and landform was also significant ($P \leq 0.05$).

Mean C content in W1 soil profiles was significantly less than the W2 soils, with W3 soils being intermediate (Table 3). At the watershed-scale, the surface 15 cm contained 32% of the total profile C in W1, 22% in W2 and 35% in W3. Thus, W2 soils have more C retained in the subsoil than in the other two watersheds.

Mean levels of profile C in the footslope and level soils were greater than the backslope, shoulder and summit soils. These greater C contents are probably due to the erosion of topsoil from the shoulder or backslope positions and deposition of sediment in the lower landscape positions. The quantity of profile C in the backslope position ($75.5 \text{ Mg C ha}^{-1}$) is less than 50% of the profile C in footslope or level soils. Similar patterns in the distribution of soil C and N among different landscape positions were also found previously in the Iowa loess hills (Aandahl, 1948; Manies et al., 2001), the Palouse loess soils of eastern Washington State (Pierson and Mulla, 1990), and at Canadian sites (Pennock et al., 1994; Gregorich et al., 1998). The distributions of ^{137}Cs and ^{10}Be in these watersheds support the deposition of sediment in the footslope and lower level positions in these watersheds (Spomer et al., 1985; Harden et al., 2002).

The landform by watershed interaction in the ANOVA was also significant, indicating different patterns of organic C distribution within the three watersheds. Therefore, the soil profile organic C contents in the different landscape positions within each watershed are shown in Table 3. The greater mean profile C content for W2 soils appears to be due to greater retention of C in the subsoil, particularly in the shoulder slope landform, but the reasons for the

Table 2
Organic carbon content (Mg C ha^{-1}) in surface soil to a depth of 15 cm for different landform elements within the watersheds

Watershed	Landform element					
	Toeslope	Footslope	Backslope	Shoulder	Summit	Mean
W1	26.9 ± 1.3^a	25.4 ± 5.9	25.3 ± 6.2	23.1 ± 5.0	25.6 ± 2.4	25.2 b
W2	28.9 ± 3.5	21.1 ± 1.9	22.7 ± 6.0	25.0 ± 4.8	25.4 ± 0.8	24.0 b
W3	32.0 ± 12.2	35.5 ± 3.4	28.4 ± 5.2	28.0 ± 3.5	38.1 ± 3.9	29.7 a
Landform mean ^b	29.1 a	27.9 ab	25.3 b	25.8 ab	28.7 ab	

^a Mean and standard deviations.

^b Differences in means were determined by analysis of variance with Type III sums of squares followed by Duncan's test. Means in the same row or column followed by the same letter are not significantly different at $P \leq 0.05$ for watershed comparisons and $P \leq 0.10$ for landform element comparisons.

Table 3

Carbon content (Mg C ha^{-1}) in soil profiles to a depth of 90 cm for different landform elements within the watersheds

Watershed	Landform element					
	Toeslope	Footslope	Backslope	Shoulder	Summit	Mean
W1	180.9 \pm 31.5	63.0 \pm 15.7	68.8 \pm 37.6	44.7 ^b	59.9 \pm 3.9	77.8 b
W2	188.4 \pm 17.6	194.6 \pm 10.8	75.7 \pm 41.3	199.7 \pm 2.7	125.1 \pm 19.2	111.4 a
W3	132.9 \pm 80.3	163.4 \pm 36.7	80.6 \pm 34.4	76 \pm 18.5	112.3 ^b	96.0 ab
Landform mean ^a	172.7 a	143.7 ab	75.5 c	112.0 bc	96.4 c	

^a Mean and standard deviations. Differences in watershed means were determined by analysis of variance with Type III sums of squares followed by Duncan's test at the $P \leq 0.05$ significance level. Means in the same row or column followed by the same letter are not significantly different.

^b Standard deviations were not computed because only one observation was present within that landform.

elevated C content in watershed W2 shoulder slope soils are not apparent.

3.3. Soil carbon and topographic indexes

Soil erosion is a process that degrades soil quality. Terrain analysis provides three secondary topographic indices, the sediment transport index, the wetness index, and the stream power index that are related to the erosion process. These differ from landform analysis through the incursion of the specific catchment area (A_s). This variable is a relative measure of the area contributing runoff to each cell. The sediment transport index is a unit-less estimate of soil erosion potential that increases proportionally to contributing area (A_s) and slope (Gallant and Wilson, 2000). It is functionally equivalent to the length–slope (LS) factor in the revised universal soil loss equation (Moore and Wilson, 1992). The index assumes rainfall is sufficient to saturate the soil profile and induce runoff uniformly over the watershed. The sediment transport

index values are greatest for the concave backslopes and in the grass waterways that carry concentrated runoff. Median values (\pm interquartile range) of the sediment transport index were 7.62 ± 8.66 for W1, 6.03 ± 7.27 for W2, and 7.01 ± 7.24 for W3 (Fig. 3). The stream power index (not shown) is a function of catchment and slope and shows where the greatest runoff water flows are likely to occur. This index is qualitatively similar to the sediment transport index.

The wetness index indicates areas where runoff water would accumulate. The wetness index increases as slope decreases and as specific catchment area increases. Although sediment is not explicitly modeled in the wetness index, sediment deposition would probably occur in areas where water is delivered onto footslopes and flow becomes slower due to decreased slope. The median values (\pm interquartile range) of the wetness index were 8.61 ± 0.97 for W1, 9.05 ± 1.02 for W2, and 8.79 ± 1.61 for W3 (Fig. 4). A comparison of the median values and interquartile ranges for the sediment transport index and the wetness index

Table 4

Correlation coefficients (r) between profile organic C (Mg ha^{-1}) and terrain parameters

Terrain parameter	Watershed W1	Watershed W2	Watershed W3
Profile curvature	−0.202	−0.296	−0.631**
Plan curvature	−0.043	−0.335*	−0.213
Slope	−0.521**	−0.711**	−0.360*
Aspect	0.364	−0.209	−0.282
Elevation	−0.667**	−0.477**	−0.080
Wetness index	0.591**	0.650**	0.445**
Stream power index	0.016	−0.331*	−0.007
Sediment transport index	−0.230	−0.591**	−0.084

* Significance at $P \leq 0.10$.

** Significance at $P \leq 0.05$.

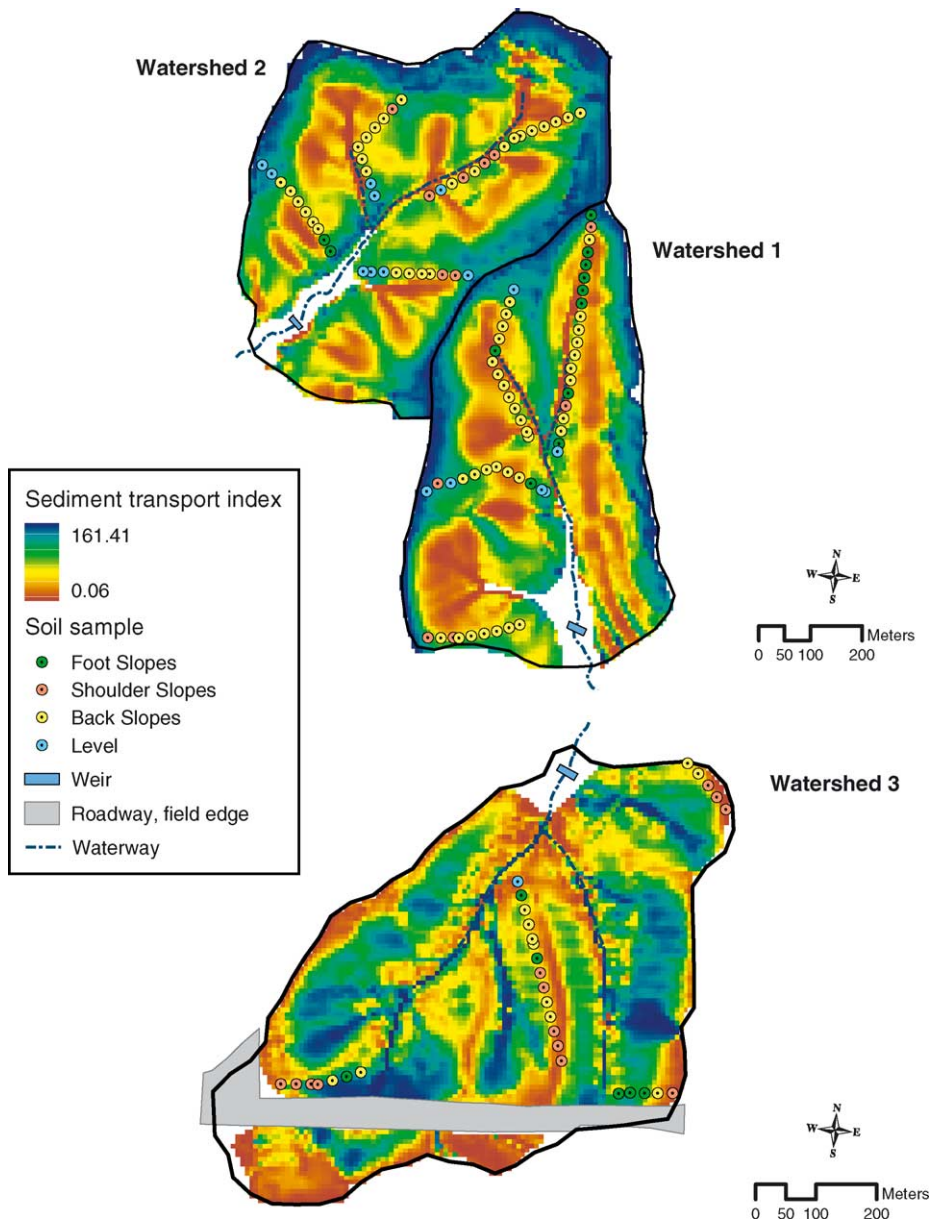


Fig. 3. Sediment transport index for three watersheds in the loess hills of southwest Iowa. Circles show the classification of sampling locations into shoulder, backslope, toeslope and level elements. Level elements were subdivided into summit (level elements at the top of the slopes) and footslopes (level elements at the base of the slopes) on the basis of elevation.

suggest that the three watersheds are similar in their terrain, although there is substantial variability within each watershed.

The correlation of profile organic C contents with these indexes and the primary terrain attributes are

shown in Table 4. In the two watersheds subjected to conventional tillage (W1 and W2), slope and elevation were the primary terrain attributes correlated with profile C content. Slope was also correlated with profile C contents in W3, but elevation was not. Slope was

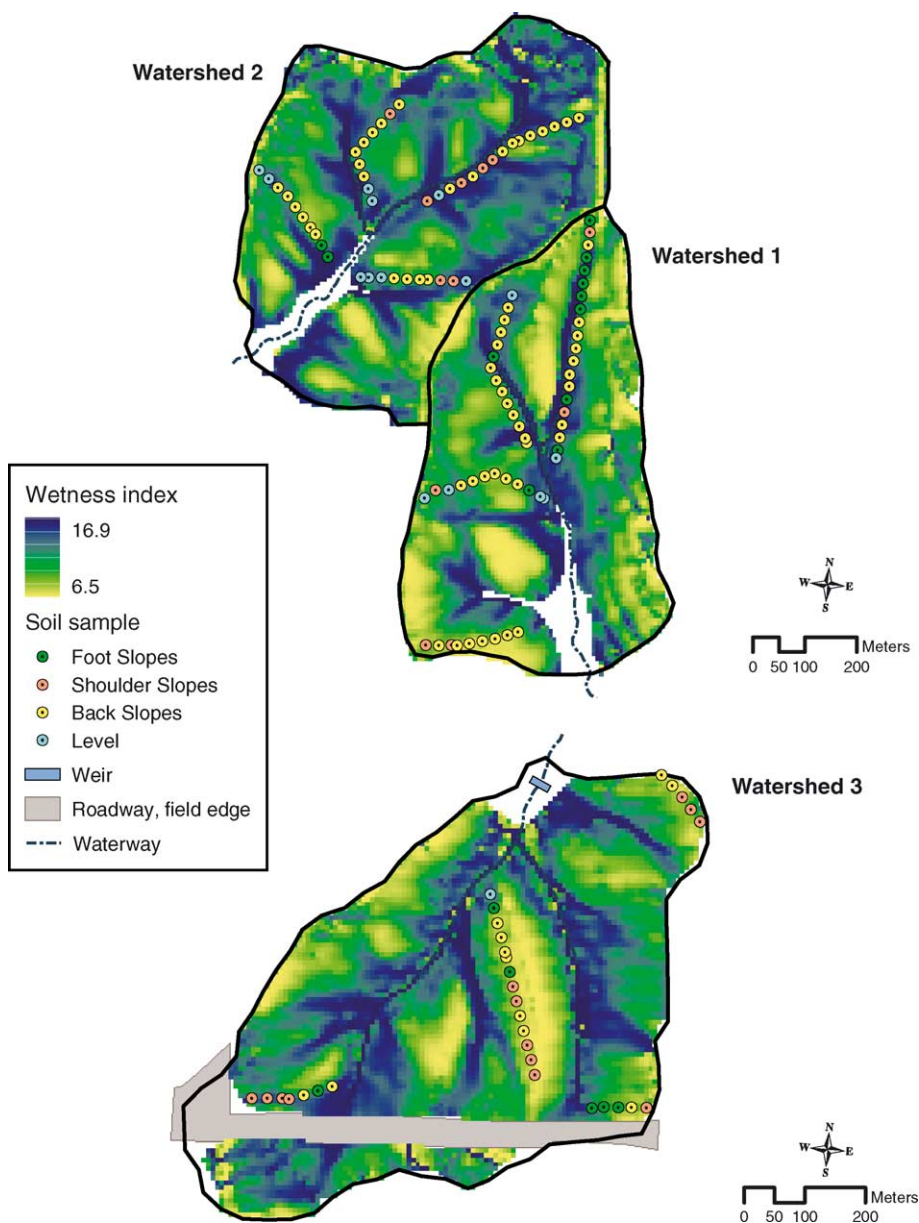


Fig. 4. Wetness index for three watersheds in the loess hills of southwest Iowa. Circles show the classification of sampling locations into shoulder, backslope, toeslope and level elements. Level elements were subdivided into summit (level elements at the top of the slopes) and footslopes (level elements at the base of the slopes) on the basis of elevation.

negatively correlated with profile C content, which is indicative of erosional loss of C from the steeper back-slopes. Elevation is also negatively correlated with profile C content. Elevation is likely significant because of the trend towards greater profile C contents

in the footslope and toeslope landscape elements. Of the three compound topographic indices, only the wetness index was consistently related to profile C contents. This positive correlation probably reflects the deposition of eroded topsoil that increase the profile

C contents in the footslope and lower level landscape elements (Table 3) where the wetness index values are higher. The backslope areas have low values for the wetness index and the lowest profile C contents. The wetness index may also be related to carbon inputs through increased plant biomass production. We did not measure C inputs directly, but the toeslope and footslope landform elements had significantly greater average corn yields than the summit, backslope and shoulder slope elements (Cambardella et al., 2004). Similar relationships between the wetness index and soil C have been found previously (Gessler et al., 1995; Moore et al., 1993).

4. Conclusions

Ridge-tillage was more effective than conventional tillage in reducing sediment loss from the watersheds. However, ridge-tillage was only equivalent to disk tillage in conserving topsoil carbon. Ridge-tillage, as practiced at this location required one or two tillage operations each year, thus its advantage in erosion protection is through the physical retention of runoff. Carbon loss through erosion was significant in the watersheds under disk tillage, but the majority of C loss was through decomposition and microbial respiration, regardless of tillage system.

Previous work suggests that tillage causes a fairly rapid decline in soil carbon, followed by a period of slower declines leading into a phase where soil C contents are in equilibrium with the production system (Gregorich et al., 1998). Watersheds W2 and W3 appear to follow this pattern, but the data suggest that W1 surface soils have been in a near-equilibrium phase since 1972. Organic C present in the subsoil exposed by erosion may also contribute to the stabilization of C content in the eroded areas, as suggested by Manies et al. (2001). Watersheds W2 and W3 now appear to be approaching a near-equilibrium phase with regard to carbon accumulation and loss in the surface soil. The different carbon dynamics between W1 and W2 suggest that management of these soils prior to this study period continues to exert influence over the carbon contents in the topsoil of these watersheds.

Watersheds are of interest for the evaluation of hydrologic processes and water quality of a given region. The importance of terrain-driven hydrologic

processes in soil formation has also long been recognized (Hall and Olson, 1991). We have shown that watersheds can be used as units to evaluate the effect of agricultural practices on soil quality parameters, but our data also show that there is considerable within-watershed variability. Backslope soils, which are more vulnerable to erosion, had less than 50% of the C present in footslope soil profiles. The footslope soils and soil in the level positions at the base of the watersheds have greater than average C contents which appears to have resulted from sediment deposition. Our watershed-scale assessments of soil quality were enhanced by the use of compound topographic indexes that appear to have the potential to identify areas within watersheds that are likely to suffer degradation through erosion. Use of these indexes may allow more precise identification of areas within watersheds that are likely to have diminished soil quality.

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